

Replacing time with space: using laboratory fires to explore the effects of repeated burning on black carbon degradation

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Abstract. Soil organic matter plays a key role in the global carbon cycle, representing three to four times the total carbon stored in plant or atmospheric pools. Although fires convert a portion of the faster cycling organic matter to slower cycling black carbon (BC), abiotic and biotic degradation processes can significantly shorten BC residence times. Repeated fires may also reduce residence times, but this mechanism has received less attention. Here we show that BC exposed to repeated experimental burns is exponentially reduced through four subsequent fires, by 37.0, 82.5, 98.6 and 99.0% of BC mass. Repeated burning can thus be a significant BC loss mechanism, particularly in ecosystems where fire return rates are high, relative to BC soil incorporation rates. We further consider loss rates in the context of simulated BC budgets, where 0–100% of BC is protected from subsequent fires, implicitly representing ecosystems with varying fire regimes and BC transport and incorporation rates. After five burns, net BC storage was reduced by as much as 68% by accounting for degradation from repeated burning. These results illustrate the importance of accounting for BC loss from repeated burning, further highlighting the potential conflict between managing forests for increasing soil carbon storage vs maintaining historic fire regimes.

Additional keywords: carbon storage, CTO-375, ecosystems, fire regimes, soil incorporation.

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Introduction

Biomass burning converts a portion of the faster cycling organic matter to slower cycling black carbon (BC) (Lehmann *et al.* 2008), which has been shown to be resistant to biotic degradation due to changes in its chemical structure (Schmidt and Noack 2000; Masiello 2004). For the boreal biome, for example, it has been estimated that >25% of carbon affected by fire is converted into a slower cycling form (Santín *et al.* 2015a). Increased fire activity is predicted for many ecosystems in response to climate change and will likely lead to significant changes in the global carbon cycle (Chapin *et al.* 2000; Westerling *et al.* 2006; Moritz *et al.* 2012; Flannigan *et al.* 2013; IPCC 2013). Given the potential for long-term carbon storage, this process has received significant attention in recent decades (Goldberg 1985; Preston and Schmidt 2006), especially given the growing interest in BC's contribution to the soil organic carbon (SOC) pool, which can exceed 50% in

some ecosystems (Lehmann *et al.* 2008; Schmidt *et al.* 2011; Cheng *et al.* 2013).

Radiocarbon dating of BC in soils has shown residence times in the order of hundreds to thousands of years (Ohlson *et al.* 2009; Egli *et al.* 2012). This variation is commonly attributed to abiotic and biotic loss processes, which can significantly shorten BC residence times (Czimeczik and Masiello 2007; Zimmerman 2010). Repeated biomass burning has been postulated to also reduce residence times of BC (Ohlson and Tryterud 2000; Rovira *et al.* 2009), but this BC loss mechanism has received less attention than most biotic mechanisms (Santín *et al.* 2013). However, in forested ecosystems that have a high propensity to produce BC through combustion of woody fuels (Hurteau and Brooks 2011), the typical time between fires is in the order of decades to centuries. This temporal scale limits the ability to observe and quantify this process in a field setting, thus emphasising the need for an experimental approach.

Black carbon cycling

Fire-altered carbon from biomass burning exists on a recalcitrance gradient, from partially charred vegetation to very recalcitrant soot (Masiello 2004; Smith and Hudak 2005; Preston and Schmidt 2006; Keiluweit *et al.* 2010). Past research into BC has been in the fields of soil science, biogeochemistry and fire science, which use differing terminology including char, charcoal, soot and graphite that can cause confusion. As described in Preston and Schmidt (2006), the term 'pyrogenically altered carbon' is used to describe the entire spectrum of carbon that has been thermally altered by fire, whereas BC often refers only to a narrow range of pyrogenic carbon produced through thermal exposure. Specifically, BC is defined here as the highly recalcitrant material that exhibits significantly lowered H:C and O:C ratios as compared to charcoal and charred biomass (Kuhlbusch *et al.* 1996; Hatten and Zabowski 2009; Brewer *et al.* 2013). Pyrogenic organic matter is another widely used term that describes BC (Santín *et al.* 2015a). Confusion may arise from use of the term 'pyrogenic carbon', which is used to describe BC (Santín *et al.* 2013) due to its closeness to the Preston and Schmidt (2006) terminology. In this study, we do not advocate the usage of a specific set of terminology, but utilise the terminology associated with the analytical BC calculation method used in this paper (Hatten and Zabowski 2009).

The coupling of BC production, soil incorporation, off-site transport and BC loss via combustion creates a complex relationship between the frequency of burning in an ecosystem and total BC storage. The molecular composition alone cannot singularly predict carbon's persistence in soils; rather, ecosystem processes including fire intensity and frequency, soil pedoturbation, and climatic fluctuations interact to play important roles in its longevity (Czimczik and Masiello 2007; Jenkins *et al.* 2014). Specifically, the physical protection of BC, for example via soil incorporation or off-site transport, is required for it to be preserved over hundreds to thousands of years (DeLuca and Aplet 2008; Fang *et al.* 2014). However, the literature describes a complex picture (Singh *et al.* 2014), in part because of the aforementioned challenges associated with different definitions of what constitutes BC. Some studies have indicated that due to soil stabilisation processes, BC levels can persist over periods of ~50 years in fallow soils (Vasilyeva *et al.* 2011). Other studies have indicated rapid losses over decadal time scales (Nguyen *et al.* 2008).

With each new fire, BC is generated, but the existing BC from prior fires may also be consumed if it remains near the soil surface and exposed to the fire (Ohlson and Tyrterud 2000; Czimczik *et al.* 2003; Rovira *et al.* 2009). Two recent field studies measured consumption of pre-existing BC (charcoal) by fire in contrasting environments. Santín *et al.* (2013) found median mass losses of <15% of BC samples placed within the surface of the organic layer consumed in a boreal forest fire; although the organic layer was consumed – exposing the charcoal – the samples survived. In contrast, when considering burying charcoal within the mineral layer Schmidt and Noack (2000) suggested that thermal degradation is not usually achieved at mineral soil depths >30 mm. Although the importance of soil insulation to prevent thermal degradation is clear, there is a lack of research quantifying soil incorporation rates of

pyrogenic carbon (Nocentini *et al.* 2010). Preliminary projections suggest that it takes decades to centuries for fire residues to be adequately incorporated into a soil matrix to ensure protection from fires (Eckmeier *et al.* 2007; Lehmann *et al.* 2008), considerably longer than fire return intervals in most dry temperate forests (Littell *et al.* 2009). Given fire frequencies in many ecosystems are expected to increase due to climatic warming (Flannigan *et al.* 2013), it is important to address how repeated burning may influence BC pools (Czimczik and Masiello 2007; Schmidt *et al.* 2011; Santín *et al.* 2015b). Saiz *et al.* (2014) evaluated fire as a BC consumption mechanism in Australian savannah woodlands, finding mass losses of <8% of BC when particles were placed on the soil surface in a prescribed fire. The low mass loss rates reported in the literature are attributed to a combination of soil insulation and fuels properties that could limit consumption (Santín *et al.* 2013; Saiz *et al.* 2014). We present results from a controlled laboratory experiment as a case study that quantifies the loss of BC biomass from an initial burn when exposed to four subsequent fire events. We hypothesise that neglecting the BC losses associated with repeated burning may lead to substantial overestimation of soil BC storage if soil incorporation does not occur to protect the particles from further pyrolytic degradation. In turn, accounting for BC losses as a function of repeated burning will lead to less BC being available for incorporation into the passive SOC pool, and these losses will likely have substantial effects on net soil BC estimates.

Methods

Sample collection and construction of fuelbeds

To test our hypothesis, we selected woody surface fuels that had been controlled for their particle size and moisture content, allowing for a controlled laboratory combustion experiment. The constructed fuelbeds represented a masticated fuel matrix (i.e. mechanically shredded and distributed tree biomass) within a western North American temperate conifer forest dominated by western white pine (*Pinus monticola*), Douglas-fir (*Pseudotsuga menziesii*), and lodgepole pine (*Pinus contorta*). Full details on sample collection, characterisation and fuelbed construction are provided in a prior study that evaluated the influence of fuel moisture on BC production rates (Brewer *et al.* 2013), and here we provide a brief summary. Fuels were sampled and collected from an 8-ha stand within the Clearwater National Forest (46°47'60"N, 119°28'12"W) that included western white pine, Douglas-fir and lodgepole pine. During mastication the wood was chipped into predominately small-diameter particles (<7.6 cm) although some larger diameter particles (>7.6 cm) were also observed (Brewer *et al.* 2013). Fifteen fuelbeds were constructed from this material to match the total surface fuel loading (5835 g m⁻²) and particle size class distribution (7.6–2.5, 2.5–1.3, 1.3–0.6 and 0.6–0.3 cm diameter, and litter) observed in the field. This total surface fuel loading value is in line with other observed fuel loadings in mesic mixed conifer systems of the north-western United States (Kreye *et al.* 2014).

Initial burn methodology

Fire experiments were conducted at the Idaho Fire Initiative for Research and Education laboratory located in a climatically

Table 1. Mean (s.d.) fuelbed characteristics and burn conditions for the five experimental burns ($n = 15$ replicates for each burn number)

Burn 1 represents the initial fire, which produced all of the >6-mm residues used in the subsequent burns represented as burns 2–5. All P -values <0.001 reported from repeated-measures ANOVA

Burn number	Bulk density (kg m ⁻³)	Fuel loading (g m ⁻²)	Consumption (%)	Fuel moisture (%)	Temperature (°C)	Relative humidity (%)
1	102.1 (9.3)	5829.7 (211.7)	90.6 (2.6)	10.0 (3.5)	17.8 (7.3)	38.4 (12.3)
2	58.7 (6.3)	2107.2 (242.6)	45.5 (13.1)	9.7 (3.9)	16.7 (2.7)	36.1 (11.0)
3	48.8 (8.4)	1771.9 (201.7)	57.4 (18.1)	11.0 (3.3)	21.5 (2.9)	34.3 (5.2)
4	45.9 (4.4)	1752.5 (97.7)	57.3 (15.4)	9.5 (4.6)	25.4 (5.2)	30.2 (6.9)
5	52.5 (16.9)	1763.3 (128.4)	61.8 (18.1)	10.2 (4.7)	21.5 (2.3)	33.6 (6.4)

controlled environment (Brewer *et al.* 2013; Smith *et al.* 2013). The experimental burn and residue collection and analysis methodologies followed the procedures detailed by Brewer *et al.* (2013). Burns were considered to be complete when mass loss had ceased, as measured using a Sartorius EB Series scale (precision: 1 g, range: 0.0005–65.0000 kg, Goettingen, Germany). Following combustion, post-fire residues were sieved into >6 mm, 1–6 mm and <1-mm size classes and weighed using a Sartorius scale (precision: 0.1 g, range: 0.1–2000.0 g), with two ~1-g sub-samples collected for BC proportion analysis. BC was quantified using thermo-chemical methods adapted from CTO375 protocols, which isolate a graphitic and biologically resistant portion of the pyrogenic carbon (Gustafsson *et al.* 1997; Masiello 2004; Hatten and Zabowski 2009; Sánchez-García *et al.* 2012). Following elemental analysis of BC proportions (CTO375_{BC}(%)), the BC mass (BC_{mass} , g) was calculated using Eqn 1 (Hatten and Zabowski 2009):

$$BC_{mass} = CTO375_{BC(\%)} * [pre_{mass} - post_{mass}] \quad (1)$$

where pre_{mass} (g) and $post_{mass}$ (g) are the original fuel mass and the mass of post-fire residues.

Repeated burns methodology

To quantify how the $post_{mass}$ residues from the initial burns persisted under subsequent burns, each burnt fuelbed ($n = 15$) was subsequently exposed to four consecutive burn trials. We acknowledge that ideally we should observe BC incorporation and losses in a field setting (e.g. Santín *et al.* 2013); however we contend that this experimental approach allowed us to simulate long fire return intervals that otherwise could only be inferred from modelling studies. In each of the subsequent burn trials a consistent litter fall was included as a layer of solely pine needles consisting proportionally (by mass) of lodgepole pine and ponderosa pine (*Pinus ponderosa*), which are common early seral species in the study region. The pine needles were added to the top of the fuelbed and selected in lieu of other litter components due to ease of replication, with the mass of needles increased to account for other missing components (e.g. leaves, twigs). These pine needle fuelbeds were constructed to resemble observed rates of litter fuel loading in the fuel collection site, which fall at the upper bound of litter fuel loading (1700 g m⁻²) in temperate conifer forests in western North America (Law *et al.* 2003; Hyde *et al.* 2011). Fuel moisture was controlled by placing prepared fuelbeds in a drying oven before combustion (Table 1; Brewer *et al.* 2013).

In each repeated burn after the initial characterisation, only the residues >6 mm in size underwent additional elemental analysis for assessing BC proportions. This component was then carried forward into the subsequent burn trials, mixed throughout the pine needle fuelbeds. Residues <6 mm were not carried forward, as these were indistinguishable from the newly burnt pine needles. To calculate percentage BC remaining [Eqn 2] in subsequent burns, BC masses were standardised against BC produced in the initial burn, by:

$$BC_{Ri} = \frac{BC_1 - BC_i}{BC_1} \quad (2)$$

where BC_{Ri} is the normalised remaining BC after burn number i (from 2–5); BC_1 is the BC produced from burn number 1 and BC_i is the remaining BC after burn number i .

Statistical analysis and modelling

A repeated-measures ANOVA was used to test for differences in post-fire residue and BC masses for each burn trial. When Mauchly's sphericity assumption was not met, the Greenhouse–Geisser statistic was used. A Bonferroni post hoc test was used to compare the main effect of burn number. To generalise our results and construct a range of partially protected BC budgets, we fit a negative exponential model, $y = a * e^{bx}$ robust to outliers, to the individual mass loss percentages of the exposed >6-mm residues, where y is the predicted remaining BC mass (% of the original mass), and x is the burn number (where $x = 1$ for the initial burn). When $a = 259.0$ (239.5 – 278.6, 95% prediction bounds) and $b = -0.9476$ (-1.008 to -0.8877), this model explained 98% of the variability in our observed data. The observed BC mass loss rate was used to develop a BC budget through the five burns. Further, a series of hypothetical BC budgets, representing varying degrees (0–100%) of protection of the total BC produced in the initial burn are produced and discussed. Statistical analyses were conducted using IBM SPSS predictive analytics software (version 19), and Matlab technical computing software (version 7.11.1). For this modelling, the BC produced in the combustion of the added litter layer is not included, as it is indistinguishable from the fine woody BC.

Results and discussion

Following the initial burns, BC biomass in the burn residues were divided into three size classes (<1, 1–6 and >6 mm), each containing significantly different proportions of the residues: ~61, 31 and 8%. The >6-mm BC from the initial burns declined

Table 2. Median (s.d.) production and loss rates associated with the post-fire residues and black carbon (BC) ($n = 15$)

Burn 1 represents the initial fire, which produced all of the >6-mm residues used in the subsequent burns represented as burns 2–5. All P -values <0.001 reported from repeated-measures ANOVA. Homogenous subsets as identified by Bonferonni post hoc analysis are identified as a, b, c, d, and e

Burn number	Residues (g m^{-2})	Residue remaining (%)	BC (g m^{-2})	BC remaining (%)
1	198.1 (77.3) a	100.0 (0.0)	0.0650 (0.0290) a	100.0 (0.0)
2	113.7 (50.9) b	55.6 (17.1)	0.0400 (0.0180) b	63.0 (23.0)
3	78.8 (46.6) c	38.5 (15.0)	0.0140 (0.0100) c	17.5 (13.4)
4	54.8 (34.9) d	24.2 (13.3)	0.0010 (0.0006) d	1.4 (0.8)
5	39.5 (30.7) e	15.7 (13.3)	0.0007 (0.0004) d	1.0 (0.6)

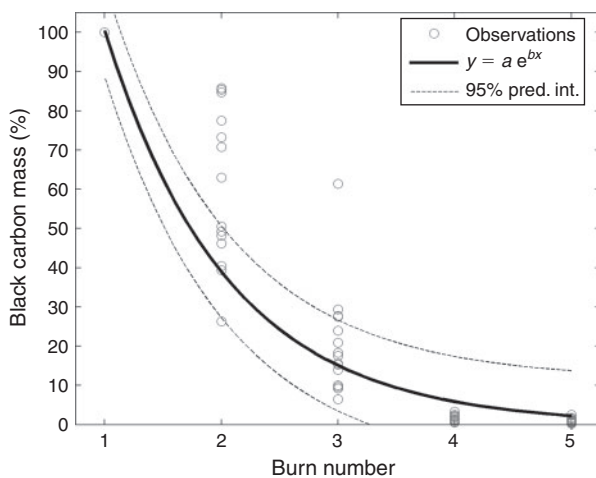


Fig. 1. Black carbon loss with repeated burning. Burn number 1 represents total carbon produced after the initial burn (100%). For each repeated burn (Burn number 2–5), 15 replicates are shown as observations. The solid black line represents a robust fit of the model $y = a e^{bx}$, and the dashed lines represent 95% prediction intervals. The fitted model explains 98% of the variability in the observations ($r^2_{\text{adj}} = 0.98$), when $a = 259.0$ and $b = -0.9476$.

significantly across the four subsequent burns (Table 2, Fig. 1). Following Burn 5, only 1% of the exposed >6-mm BC produced in the initial burn (Burn 1) remained. Integrating the charred residues throughout the litter layer probably made the charred particles more susceptible to thermal degradation, as the thermal insulation (e.g. provided by being at the bottom of the litter layer or within the O-horizon – soil layers with high amounts of organic matter) would have been reduced (Santín *et al.* 2013). However, in a field setting, charred residues could remain at the interface of the O-horizon and mineral soil, or the material could be mixed throughout the O-horizon; depending on the pedoturbation intensity and soil texture, eventually the char will become incorporated into mineral soil (Gavin 2003). Consequently, our estimates are likely to represent an upper bound of losses associated with repeated burning, and our results highlight the need for future research on pedoturbation within the O-horizon and influence of soil textures on BC soil incorporation. Further, use of the CTO375 method of characterising the quantity of BC sets a fairly rigid definition that excludes part of the BC continuum, especially were BC is produced under low charring temperatures (Baldock and Smernik 2002; Chen *et al.*

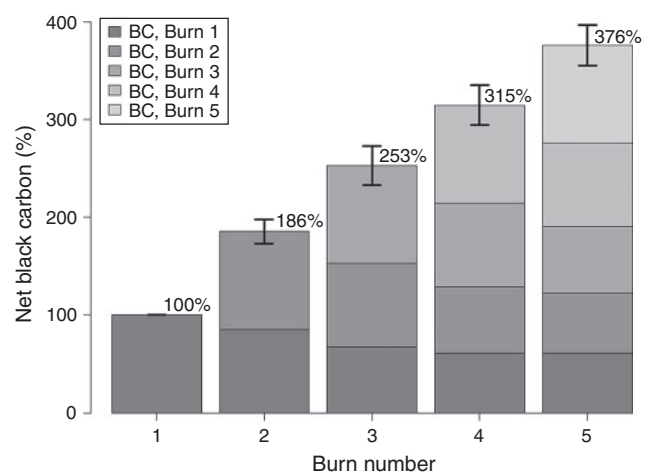


Fig. 2. Black carbon (BC) budget based on the percentage of BC remaining, reported in Table 1. The net BC contributing to the budget of each burn number is tracked via varying shades of grey. Each burn contributes 100% of the BC generated in the initial burn; 61% of this BC is protected from degradation and 39% is considered exposed through subsequent burns following the observed loss rates.

2014). Nonetheless, our results support the prevailing hypothesis that in forest types with high-frequency burning and little to no soil incorporation, repeated burning can be a significant mechanism for BC loss (Ohlson and Tryterud 2000; Preston and Schmidt 2006; Rovira *et al.* 2009). Our results also indicate that in as few as two repeated burns, the majority (~80%, Table 2, Fig. 1) of the exposed BC produced in an initial fire can be lost. Given that the best estimates of BC loss rates through biotic and abiotic (non-pyric) mechanisms range from <1% to 37% over 100 years (Zimmerman 2010), our results highlight repeated burning as a potentially significant mechanism of carbon loss.

To estimate the compounding effects of BC loss through repeated burning, we used the experimental loss rates of the exposed carbon (Fig. 1) to construct a BC budget spanning five burns (Fig. 2; Ohlson and Tryterud 2000; Czimczik and Masiello 2007; Zimmerman 2010). This budget assumes all material <1 mm is protected from future burning, implicitly representing immediate off-site wind transport, as is common in grassland and savannah ecosystems, or incorporation of fine particles into the soil and litter matrix (Rumpel *et al.* 2009). Protecting the <1-mm BC left 39% of the BC produced in an

initial burn available for further thermal degradation. This scenario also assumes that an equivalent masticated fuel loading was reached between each burn, representing a best-case estimate for dry forest types, where net BC increases at a near-linear rate of ~70% per burn (Fig. 2). By the fifth burn, BC storage is 381% of that created in the initial burn, whereas an estimate ignoring BC loss from repeated burning would predict a value of 500%. The ‘missing’ 119% represents the tradeoff between generating new BC while consuming existing BC with each successive burn. We acknowledge that a full accounting of the residual size categories would provide more accurate estimates of BC loss rates; however, given analytical limitations of separating these fine char fractions from pine needle residues in the repeated burns, this was not feasible. As grass fires likely produce charcoal residues smaller than 6 mm, the conclusions of this experiment are limited to ecosystems with woody vegetation (i.e. trees and shrubs), which represents a continuum from savannah to mesic forest.

The degree to which repeated burning influences a BC budget inherently reflects assumptions on BC protection from future burning (e.g. via soil incorporation or off-site transport; Rumpel *et al.* 2009; Dittmar *et al.* 2012; Santín *et al.* 2013). To generalise our results and explore their sensitivity, we used the fitted model in Fig. 1 to calculate net BC storage under five different scenarios, where 0%, 25%, 50%, 75% and 100% of the BC produced in the initial burns is protected from future burning (Fig. 3). As expected, BC budgets are highly sensitive to protection rates: at 0% protection, net BC asymptotes around 160% after four burns, implying an upper limit to BC storage. Under the scenario with 75% BC protected, values fail to asymptote and reach 416% following the fifth burn (Fig. 3).

Implicitly, these scenarios are both sensitive to, and represent a series of, generalised environmental conditions that dictate both BC protection rate (% per year) and the rate of burning (fire per year) in a given ecosystem. For example, although savannah ecosystems are frequented by fires every 3–5 years, mesic forests can exhibit fire return intervals of 200+ years. Although less well known, soil incorporation and off-site wind transfer rates of BC in these systems likewise vary (Rumpel *et al.* 2009; Nocentini *et al.* 2010; Kasin and Ohlson 2013), principally due to differences in soil exposure, topography, precipitation, temperature and wind regimes.

We posit that if burning rates are faster than the protection rate required to safeguard BC, then most BC produced in a burn will still be exposed during subsequent burns. This may represent a scenario of 0% or 25% BC protection between burns, which may be comparable to the short fire return intervals found in pine–savannah systems if off-site transport through wind and water erosion is minimal; this scenario may not be commonplace, given these particles are often considered very susceptible to off-site transport by wind and water processes in grasslands and savannahs. In contrast, if similar tradeoffs were observed in other systems such as the longer fire return intervals common to boreal forests, such systems may exhibit enough time between burns for BC to be incorporated sufficiently far into the litter and duff or mineral soil to be protected during subsequent burns. Such ecosystems are more likely represented by a scenario of 75% or 100% BC protection (Santín *et al.* 2015b).

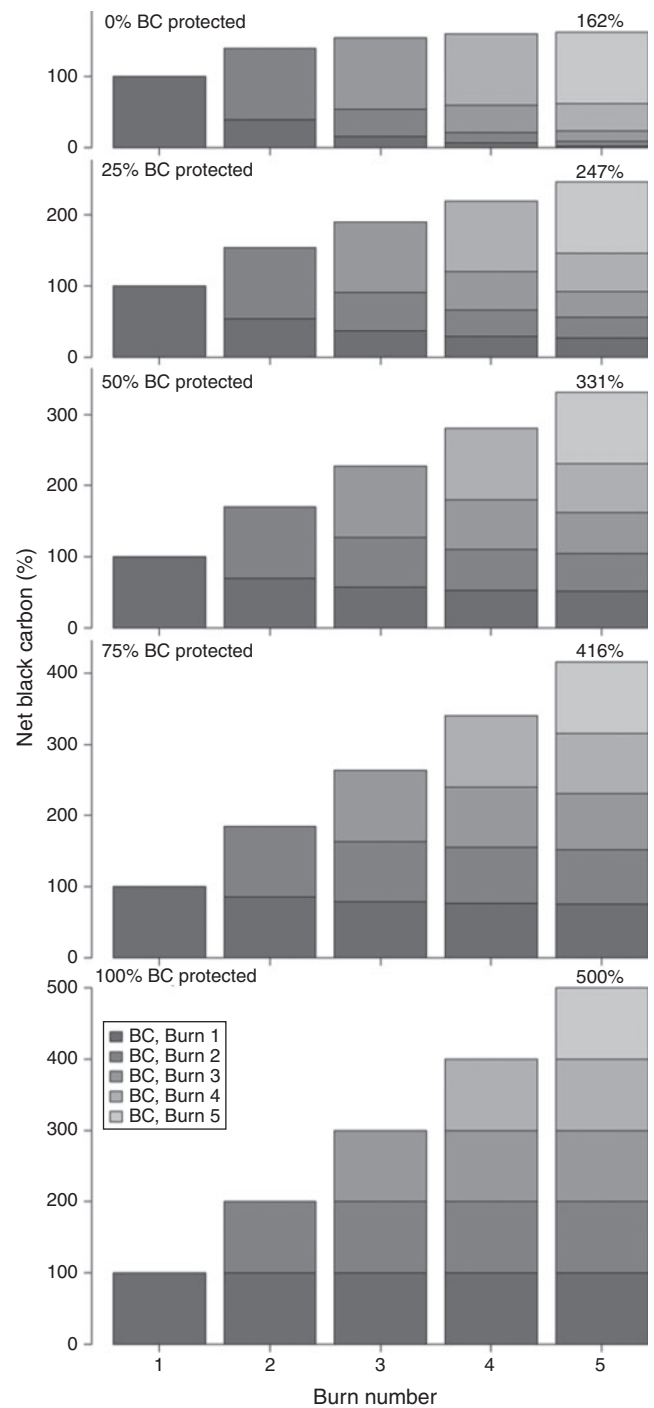


Fig. 3. Black carbon (BC) budgets based on varying modelled scenarios; each budget assumes that a set level of BC, varying from 0–100% in 25% steps, is protected from subsequent burning. The remaining exposed BC follows the fitted model in Fig. 1 to calculate loss rates. Each scenario implicitly represents varying soil incorporation rates and fire frequencies across a range of fire-prone ecosystems.

We contend that ecosystem variations affecting the ratio of these two processes (fires per percentage BC protected), that is, the rate of burning (fires per year) and rate of BC protection (% per year), could determine the sensitivity of soil BC budgets

to repeated burning. Assessment of this ratio could enable studies to determine if and when net BC storage reaches an asymptote, but will require understanding the protection and production rates of residues of different sizes in different ecosystems. If frequent fire consumes BC faster than it can be protected by soils or off-site transport, then those forested systems with short fire return intervals would have lower amounts of BC compared with forested systems with long fire return intervals. However, the per-fire production rate of black carbon is a function of other factors such as fire severity and biomass production between fires, which complicate studies trying to determine the variability in this relationship (Jauss *et al.* 2015).

Ultimately, these processes likewise dictate the feasibility of using fuel treatments and fire hazard management (prescribed, wildland fire use fires, etc.) as tools to increase soil BC storage (DeLuca and Aplet 2008; Santín *et al.* 2015b). Given the tradeoff between BC production and consumption in subsequent burns, maintaining 'natural' fire regimes in ecosystems historically characterised by high-frequency fires (e.g. burning once every several years to decades), may be at odds with maximising soil BC storage (Cheng *et al.* 2013).

Conclusion

This study highlights the potential importance of physical loss of BC through repeated burning, adding combustion as a key mechanism to previous work demonstrating BC loss through biological and physical degradation. For BC that remains in situ to be most effective as a net carbon sink, it must be incorporated into the mineral soil matrix before subsequent burning. Our work is a first step towards quantifying BC loss rates from repeated burning to more accurately model long-term BC storage in soil organic pools, but projecting the long-term impacts on carbon budgets requires more precise estimates of BC protection rates. Biogeochemical models that track BC should be sensitive to the combined effects of burning, soil incorporation and off-site transport rates, as exemplified by the ratio of these processes.

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